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Life cycle analysis approach to comparing environmental impacts of alternative materials used in the construction of small wastewater treatment plants

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ABSTRACT

With the aim of reducing the environmental burden of decentralized wastewater treatment plants in India, this project investigated five primary materials (stainless steel (SS), mild steel (MS), glass fibre reinforced polymer (GFRP), high density polyethylene (HDPE), and reinforced concrete cement (RCC)) in terms of the relative environmental impact that each would incur across 13 midpoint and 4 endpoint impact categories during the early life stages. The results showed that SS demonstrated substantially higher impact in total (5.47 Pt) and across each of the endpoint categories, most notably human health (3.12 Pt). Further investigations demonstrated that this was largely fed by the respiratory inorganics midpoint category that accounted for 50 % of the total impact (2.75 Pt), while global warming (0.93 Pt), non-renewable energy (0.70 Pt) and terrestrial ecotoxicity (0.62 Pt) were the only other considerable impacts. GFRP incurred the second greatest impact overall (2.32 Pt), while MS, RCC and HDPE followed with 1.82 Pt, 0.78 Pt, and 0.39 Pt respectively. HDPE afforded the greatest efficiency in all midpoint categories except carcinogens where RCC incurred the least environmental cost. Results were then compared with previous work and likely causal factors highlighted. Further study is recommended to investigate the longevity of the alternative materials in a wastewater containment role to support these results.

1. Introduction

As world leaders pledge to cut emissions and reduce environmental impact, greater focus is being given to the sustainable development of infrastructure to realise these gains (Arce and Gullón, 2000; Mirza, 2006; Doyle and Havlick, 2009; Zayed et al., 2011; UN General Assembly, 2015; Battacharya et al., 2020). Perhaps most critical is ensuring the availability of water and sanitation to all as targeted by the United Nations (UN) under the 17 Sustainable Development Goals (SDGs) established in 2016 (UN General Assembly 2015). With clean water and sanitation now officially recognised as a human right by the UN General Assembly, global momentum has been gaining to supply these services to those still lacking these basic facilities (World Health Organization, & United Nations International Children's Emergency Fund 2013; WHO, 2015; Cha et al., 2017). Despite this, the World Health Organization (WHO) suggests a quarter of the world's inhabitants still lack safe sanitation indicating significant amounts of water infrastructure is still needed (WHO, 2019). If this SDG is to be achieved by 2030 as targeted, then an environmentally-sensitive approach to its implementation will be necessitated.

India is recognised as a priority country for improved coverage of sanitation, accounting for much of the world's deficit in sanitation (Coffey et al., 2015; Nandi et al., 2017). Despite government reports that Mohdi's 5-year Clean India Mission had now successfully provided latrines to 95 % of households (National annual rural sanitation survey NARSS 2018-19. Government of India), independent assessments have reported a lack of adoption by communities due to poor quality and inadequate maintenance plans that may lead to overflow and increased sewage exposure (Coffey et al., 2015; Exum et al., 2020; Versano, 2020). Coverage in urban areas remains divided by social-economic factors (Cha et al., 2017; Saroj et al., 2020), while the negative health effects and mortality in children due to poor sanitation are exacerbated by high population density (Hathi et al., 2017; Augsburg and Rodriquez-Lesmes, 2018). Even before environmental considerations, coverage of effective sanitation continues to be thwarted by financial and circumstantial constraints (Wilderer, 2005).

Decentralization affords a plausible solution for overcoming key challenges of implementation in India with reduced environmental impact (Wilderer, 2005; Massoud et al., 2009; Starkl et al., 2012; Brunner et al., 2018). Economics and environmental impact are intrinsically linked when one considers the cost of each during

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sewage pipe installation, which is avoided with end-of-pipe treatment (Wilderer, 2005). Maintenance and operational costs are also substantially reduced in comparison (USEPA, 1997), while wastewater reuse and resource recapture are better facilitated (Parkinson and Tayler, 2003; Nanninga et al., 2012; Tchobanoglous and Leverenz, 2013). Due to the versatility of decentralized treatment, it continues to be advocated as a favoured strategy in developing countries for providing sanitation to peri-urban areas (Parkinson and Tayler, 2003; Beausejour and Nguyen, 2007; Nanninga et al., 2012; Brunner et al., 2018), densely-urban areas (Opher and Friedler, 2016; Kuttuva et al., 2018; Reymond et al., 2020) and small rural communities (Galvão et al., 2005; Singh and Kazmi, 2018). It is then unsurprising that decentralized wastewater treatment has been gaining such momentum in India as a promising resolve for the sanitation crisis (Singh et al., 2015; Singh and Kazmi, 2018).

Life cycle analysis (LCA) provides a means to further analyse environmental impact by investigating the impact across a technology's life cycle. With regards to sanitation, this approach has been used widely as a decision-making tool to compare the total environmental burden of different decentralized technologies (Machado et al., 2007; Nogueira et al., 2009; Opher and Friedler, 2016), the impact of different strategies during individual life phases of the treatment plant (Singh et al., 2017, 2020) and to identify the most costly phases of the life cycle (Vahidi et al., 2015; De Feo et al., 2016; Morera et al., 2017).

This paper aims to investigate the use of several alternative materials and their respective processes that may be used as primary materials during construction of a small, decentralized wastewater treatment plant (WWTP) in India, and the relative environmental costs that each material can incur in that role. While LCAs remain the most commonly used method for evaluating the environmental impact of WWTPs, the construction phase continues to be underrepresented in life cycle investigation (Remy and Jekel, 2008; Corominas et al., 2013; Morera et al., 2017; Gallego-Schmid and Tarpani, 2019). This was emphasized by Corominas et al. (2013) in their review of LCA use with regards to WWTPs who found less than half of the reviewed work accounted for the construction phase, while a critical review of WWTP LCAs by Gallego-Schmid and Tarpani (2019) found this portion to be even lower.

Of the studies that have included the construction phase in their investigation of sanitary infrastructure, its influence as one of the most costly phases of the lifecycle has been repeatedly highlighted (Vahidi et al., 2015; De Feo et al., 2016; Morera et al., 2017; Singh et al., 2017, 2020). Vahidi et al. (2015) observed the production phase to be the most heavily impacting phase of the sewage pipeline life cycle. De Feo et al. (2016) found the construction phase of WWTPs to be the most costly phase of the life cycle after the operational phase. Similarly, Singh et al. (2017) compared the environmental impact of a small WWTP across the construction and operation phases, and found the former to be the most impacting in terms of toxicity indicators. A detailed LCA by Morera et al. (2017) concluded that the construction phase accounted for over 5 % of the environmental impact of an energy-intensive activated sludge (AS) plant over its life span and as much as 60 % for metal depletion. In less energy-intensive systems, the construction phase showed accountability for 67 % of the environmental impact compared to only 33 % for the operational phase (Lutterbeck et al., 2017).

Material choice is a key influence on the relative impact of the early life stages, i.e. construction phase, (Shah et al., 2016; Singh et al., 2017; Burchart-Korol and Zawartka, 2019; Singh et al., 2020). In their study, Singh et al., (2017) identified the use of stainless steel (SS) to be the major contributor across the various endpoint impact categories (i.e. human health) during the WWTP construction, concluding that the use of alternative materials for tank construction could generate substantial sustainability gains. These findings were supported by a follow-up study which investigated the mid-point categories (i.e. respiratory inorganics) and also emphasised stainless steel (SS) to be a heavy but avoidable environmental cost (Singh et al., 2020). Despite this, material comparisons from a LCA perspective in the WWTP role are lacking in the literature.

While previous work has attempted to compare environmental impact associated with material choice in the WWTP role, these studies have either focused on only a few select impact categories or on assets that do not represent the disproportionate material quantities required to contain higher volumes of wastewater in line with ISO structural standards. For example, Machado et al. (2007) investigated the use of alternative materials in larger structures such as activated sludge (AS) reactors, however their findings were limited to only CO_2 emissions and abiotic depletion. While other examples have investigated the impact of different materials in a broader range of categories but in wastewater pipes (Vahidi et al., 2016) or municipal solids waste management (MSWM) systems (Rives et al., 2010) that are not comparable in terms of material quantity.

Environmental impacts pertaining to the use of SS in construction are well documented (Palaniappan & Karthikeyan, 2009; Cena et al., 2015; Dunea et al., 2016; Usman et al., 2019). The alloy that constitutes SS is characterised by many different elements whose quantities vary depending on the type of SS (i.e. austentic, ferritic, duplex, martensitic). These elements include several heavy metals such as chromium (Cr), nickel (Ni), and copper (Cu) to name a few. While the addition of these elements is known to improve resistance to corrosion, heat, and bio-foul (Yellishetty et al., 2011), they also demonstrate a high level of toxicity that can be highly detrimental to human and ecosystem health (Palaniappan and Karthikeyan, 2009; Cena et al., 2015; Dunea et al., 2016; Usman et al., 2019). This highlights the need to identify alternative materials that may be suitable for the role of wastewater containment at reduced environmental costs in line with international pledges.

2. Methodology

2.1. Software and analysis methods

In order to investigate the environmental load of the alternative materials considered, a commercially available LCA software (SimaPro PhD 8.5.2) was used. This tool is used widely in the manufacturing sector as a way to assess environmental impact and its application in water sector is reported (Lundin et al., 2000; Machado et al., 2007; Vahidi et al., 2015; Singh et al., 2017).

A life cycle impact assessment (LCIA) was carried out to investigate the construction phase of the life cycle using the IMPACT 2002+ damage assessment method for comparability with previous work (Singh et al., 2020). This method was used by Singh et al. (2020) on the same small WWTP.

IMPACT 2002+ is a combination of four methods including IMPACT 2002, Eco-indicator 99 (2nd version, Egalitarian Factors), CML and IPCC (Jolliet et al., 2003). It links life cycle inventory results to four endpoint damage categories (human health, ecosystem quality, climate change and resource use) by way of midpoint categories. An example of the distinction between midpoint and endpoint categories would be that midpoint quantifies ozone depletion potential while endpoint measures skin cancer or crop damage as a result of increased ultraviolet B-rays (UVB) radiation due to ozone depletion. The 13 midpoint categories included in the analysis are; human toxicity, respiratory effects, ionizing radiation, ozone layer depletion, photochemical oxidation, aquatic ecotoxicity, terrestrial ecotoxicity, terrestrial acid/nutrition, land occupation, global warming, non-renewable energy and mineral extraction. These midpoint indicators were used to characterize the elementary flows as well as other environmental interventions that contribute to a common impact (Jolliet et al., 2003). Resource use and the environmental emissions associated with the product under investigation were quantified as well as the relative contribution to each of the potential impact categories (Hischier et al., 2010).

Within the IMPACT 2002+ method, multiple indices and units are used. The Pt unit is a dimensionless ecological value, where every Pt indicates 1000th of the yearly environmental load of one average European inhabitant (Hischier et al., 2010). The disability-adjusted life year (DALY) index is used when an impact concerns human health, and is a way of quantifying the overall disease burden of an impacting factor, expressing the number of years lost due to ill-health, disability or early death (Hischier et al., 2010). Damage to ecosystem quality is measured in potentially disappeared fraction of species times the area over which they disappear times the number of years of damage (PDF*m^{2*}yr). Finally MegaJoule (MJ) surplus is used as an indicative measure of resource scarcity, or by definition, the total additional future cost to the global society due to the production of one unit of resource (Hischier et al., 2010).

2.2. Study system

The small WWTP that this scenario is based on is an integrated fixed-film activated sludge (IFAS) technology as examined in previous studies (Singh et al., 2017, 2020). IFAS systems provide an effective decentralized solution particularly in areas of limited land availability as is common in built up areas. This is due to their capacity to hold larger amounts of functional bacteria than conventional activated sludge plants due to the inclusion of fixed media in the reactor that promotes biofilm growth in addition to the suspended colonies (Singh and Kazmi, 2016). The IFAS system considered consists primarily of an aeration tank with dimensions 3 m length x 2 m width x 3.34 m height (total volume = 20 m^3) and a 3.34 m high settlement tank of cylindrical design with a conical bottom (total volume 4.2 m^3). It is considered that both these tanks are constructed using the same material.

2.3. Goal and scope descriptions

The goal of this life cycle study was to compare alternative primary materials that may be used in the construction of a small WWTP treating municipal wastewater in order to identify potential savings in environmental cost. Mild steel (MS), high density polyethylene (HDPE) plastic, glass fibre-reinforced polymers (GFRP), and reinforced concrete cement (RCC) were investigated being commonplace materials used during construction of different assets from pipe networking to treatment reactors (Marsh, 2009; Vahidi et al., 2015). GFRP is becoming more widely used, not only in water and sewage applications, but as a replacement for the storage of highly corrosive substances such as fuel (Marsh, 2009; Kumarasamy et al., 2019). Plastic polymer based materials such as HDPE also offer great advantage as cheap and lightweight alternatives to steel and concrete for pipe networks and small-scale wastewater systems due to their inert characteristics (Li-xia, 2007; MortezaNia and Othman, 2012; Petit-Boix et al., 2016; Sangwan and Bhakar, 2017). Scenarios 1-5 represented SS 316, MS, GFRP, HDPE and RCC respectively.

2.4. Experimental design

While Scenarios 1, 2 and 5 follow the given design specifications, Scenarios 3 and 4 follow a different format. Scenario 3 represents the construction of the small WWTP using pre-designed, square GFRP panels (1 m x 1 m). It was therefore impractical for the settlement tank to be designed as a cylinder with conical base. Instead, the complete system was designed as a single rectangular tank with two chambers (IFAS chamber; 3 m x 2 m, Settlement chamber; 1 m x 2 m) separated by a baffle plate. Regarding Scenario 4, the authors were unable to identify safety standards that could guide the design of a rectangular or a cylindrical tank orientated horizontally. Instead both main tank and settlement tank were each designed as vertically-orientated cylindrical tanks according to ASTM D 1998 06 guidelines.

Within the supplementary material a full breakdown of the material distribution and process data in scenarios 1-5 can be found in Tables S1 and S2 respectively, while all calculations can be observed in Section S2 of the supplementary material. With regards to Scenario 1, material quantities were based upon previous work (Singh et al., 2017, 2020). In these papers, the authors described in detail all elements of the small WWTP that is central to this investigation, however, only the tank shell materials including 3,500 kg of SS 316 for the reactor and a further 400 kg for the settlement tank are relevant to this study. This value for the main reactor was known to include other brackets, frame and fixtures of the same material which were not part of this investigation so for this reason the value was recalculated. Material requirements in Scenario 2 were assumed to be the same as Scenario 1 in terms of weight due to the same density of both materials and both having sufficient mechanical properties for the role (Howard, 2003), however further processes are necessitated to overcome the limitations of MS compared to SS 316 as detailed in Table S2.

Both tanks were constructed of the same material to reflect the growing market of package-type small WWTPs in India, whereby all included system chambers (i.e sedimentation, anoxic, aerobic etc) are contained within a single transportable unit. While the IFAS system consists of many other components and materials including foundation, piping, pumps, media etc, these were discounted from the LCA to increase resolution of the analysis as practiced in previous LCA (Joshi, 1999). These materials were generic across scenarios and while known to be influential in whole system LCA did not contribute to the investigation in hand (Morera et al., 2017). A more detailed description of these components including their materials can be found in previous studies where their relative impacts were assessed (Singh et al., 2017, 2020).

This study considered a number of processes within the analysis. Every steel used was considered to have been rolled, while the MS was assumed to have undergone powder coating to 80 μm thickness to overcome the lacking anti-corrosive properties afforded by SS 316. Furthermore, all seams on each of the steel tanks were assumed to have been gas welded using acetylene. Relative quantities involved in each process are displayed in Table S2.

GFRP panels may be manufactured in a number of ways including hand lay-up, spray-up, vacuum bag moulding, resin infusion, autoclave moulding and compression moulding (Anderson et al., 2004). For this investigation the GFRP panels were considered to be manufactured by hand lay-up. HDPE water tanks of this size are typically fabricated by way of rotational moulding, however this process was also unsupported by Simapro software. In this case inventory data was input manually following indication by De Feo et al. (2016) that rotational moulding requires 3 MJ of natural gas per kg of polyethylene (PE) moulded.

2.5. System boundaries and functional unit

In this study, the system boundary considers the construction phase of the IFAS system (IFAS reactor and clarifier) that can maintain effective operation for a 15 year lifespan. The 15 year time period was chosen in line with past studies due to this being the expected lifespan regardless of structure and material used (Emmerson et al., 1995; Lundin et al., 2000; Vlasopoulos et al., 2006). The system boundaries were defined as according to Fig. 1 and includes material extraction, energy consumption, resources used in production, material transportation and system manufacture within the analysis.

The functional unit is a measure of performance of the system under investigation and provides a reference by which the results may be compared with similar studies (Vlasopoulos et al., 2006). For this study the functional unit was considered to be 24.2 m^3 of contained wastewater under aerobic treatment for 15 years.

3. Results and discussion

3.1. Comparison of total impact contribution from each scenario

Investigation into the different midpoint impact categories by way of the embodied Eco-invent 99 method permitted comparison of the impact between each scenario. This was represented in two ways. The first representation is damage assessment shown in Fig. 2a, which portrays

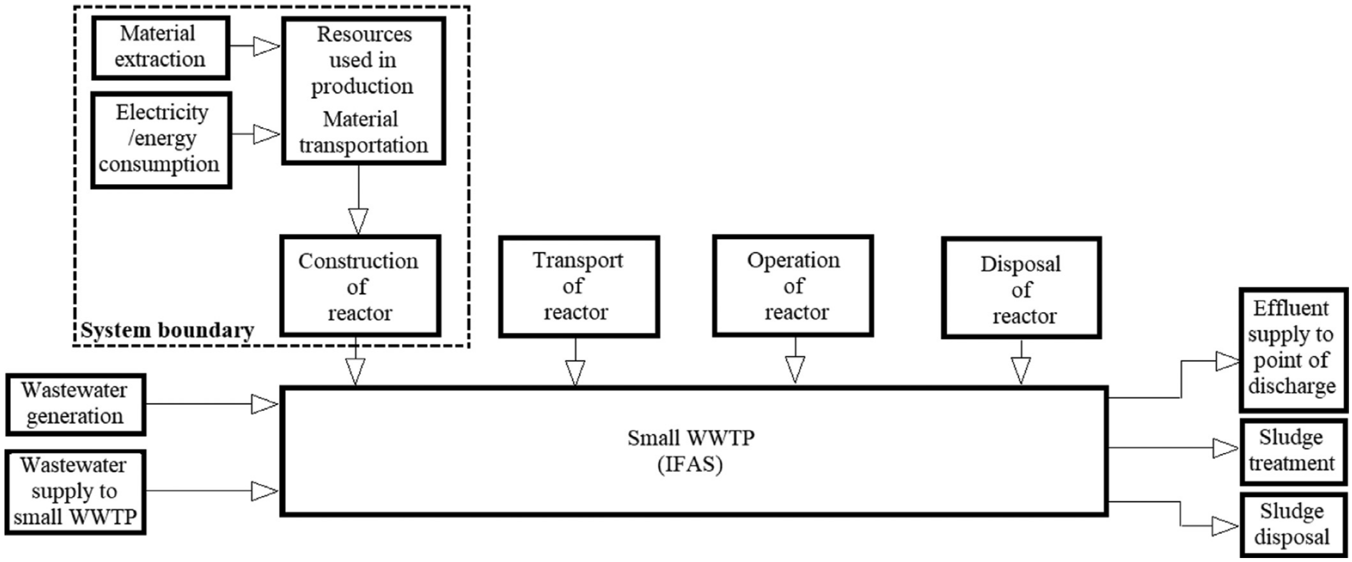

Fig. 1. System boundary schematic of the present LCA. (TIFF file * 2 columns).

the impacts of the scenarios in each of the categories. While the representation in Fig. 2a is useful for identifying which scenarios are the most impacting in individual categories, it does not portray the relative contribution of each category with regards to the total impact. The second graphical representation is the single score representation as presented in Fig. 2b. This affords a visual assessment of the inter-scenario and intra-scenario contributions to each midpoint category, which is advantageous for identifying the most impacting categories in each scenario. A limitation of this representation is that resolution of information regarding the lower contributing categories is lost and places emphasis on the need to inspect both in tandem.

Fig. 2a showed that Scenario 1 incurred the greatest impact in 8 of the 15 categories which suggests SS not only incurs the greatest amount of impact, but is also the most harmful to the environment in more ways than alternative materials. In comparison, Scenario 3 demonstrated the second broadest impact range with a total of 5 categories. Scenario 4 demonstrated the lowest impact in 11 of the midpoint categories, while Scenario 5 demonstrated the second lowest impact in all assessed categories except carcinogens where it incurred the least impact. Aquatic acidification and aquatic eutrophication categories show empty values due to no associated endpoint category (Jolliet et al., 2003).

While HDPE was observed to be the superior material across most midpoint categories compared to other scenarios, this was in contrast to a previous study by Rives et al. (2010). In their study MS was seen to outperform HDPE in all 8 of the mid-point categories investigated when different materials were compared in the construction of MSWM systems. However the study by Rives et al. (2010) considered the HDPE to be a raw virgin material while MS was produced from 40 % recycled steel and took into account the increased longevity of MS over HDPE in that role. With the raw materials stage being responsible for 60 % of the impact for HDPE and 80 % of the MS, this helps explain much of the disparity with the present study

GFRP was shown as a preferable option to SS in the majority of categories except carcinogens, non-carcinogens, ozone layer depletion, respiratory organics and land occupation. MS showed environmental gains in all categories compared to SS but demonstrated a greater impact than GFRP in several categories including respiratory inorganics, aquatic ecotoxicity, terrestrial ecotoxicity and mineral extraction where MS demonstrated a greater impact. This supports a recent study by Işildar et al. (2020) who compared the use of GFRP and structural (mild) steel in rebar production and identified GFRP to have a broad impact across midpoint categories that were in good agreement with the present study. Similarly, Vahidi et al. (2016) identified GFRP to demonstrate the greatest impact across the same range of midpoint categories when compared to RCC and HDPE, however neither steel type was included in their study.

Together Fig. 2a and 2b shows that the use of HDPE in place of SS will afford environmental benefits across most impact categories with the exception of non-renewable energy which RCC would afford the most gains. HDPE is known to be heavy on non-renewable energy in the early stages of the life cycle due primarily to material production processes (Sangwan and Bhakar, 2017). While Fig. 2b identifies non-renewable energy to be the primary impact in the midpoint categories, Fig. 2a suggests this is still far lower than SS and to a lesser extent GFRP. In contrast, a study by Burchart-Korol and Zawartka (2019) compared the construction phase of sceptic tanks in Poland made of different materials and found the amount of HDPE used to be the key indicator of impact in terms of non-renewable energy (23%) compared to steel (21%), concrete (18%) and polyester resin (15%).

Fig. 3 represents the results in terms of the endpoint categories and shows that Scenario 1 demonstrated a substantially higher total impact than the other scenarios in most damage categories. Total impact scores were observed as 5.47 Pt under Scenario 1 followed by 2.32 Pt under Scenario 3, 1.82 Pt under Scenario 2, 0.79 Pt under Scenario 5 and finally 0.386 Pt under Scenario 4. This gave an initial indication that in the early life stages SS is a substantially less sustainable material than alternatives, such as RCC that demonstrated 85.6 % less impact than SS and most notably HDPE with 92.9 % less impact. MS incurred 66.7 % less damage than SS, while GFRP demonstrated potential impact savings of 57.6 % as an alternative material.

The environmental benefits of replacing SS with HDPE for the storage of corrosive liquids has previously been identified (Stephens et al., 1998; Joshi, 1999). Stephens et al. (1998) carried out a comparative LCA of HDPE and SS in the vehicle fuel tank role and found HDPE to incur substantially less environmental impact compared to SS. A similar study by Joshi (Joshi, 1999) supported their results by also finding SS to incur the greatest environmental costs compared to plastic in vehicle fuel tank production. In their comparison of septic tank materials, Burchart-Korol and Zawartka (2019) found HDPE to outperform SS in most impact categories.

The relative contribution of each scenario to the endpoint categories was then considered. With regards to Scenario 1, human health incurred the highest impact of all the categories with a score of 3.12 Pt followed by climate change (0.93 Pt), resources (0.76 Pt) and ecosystem quality

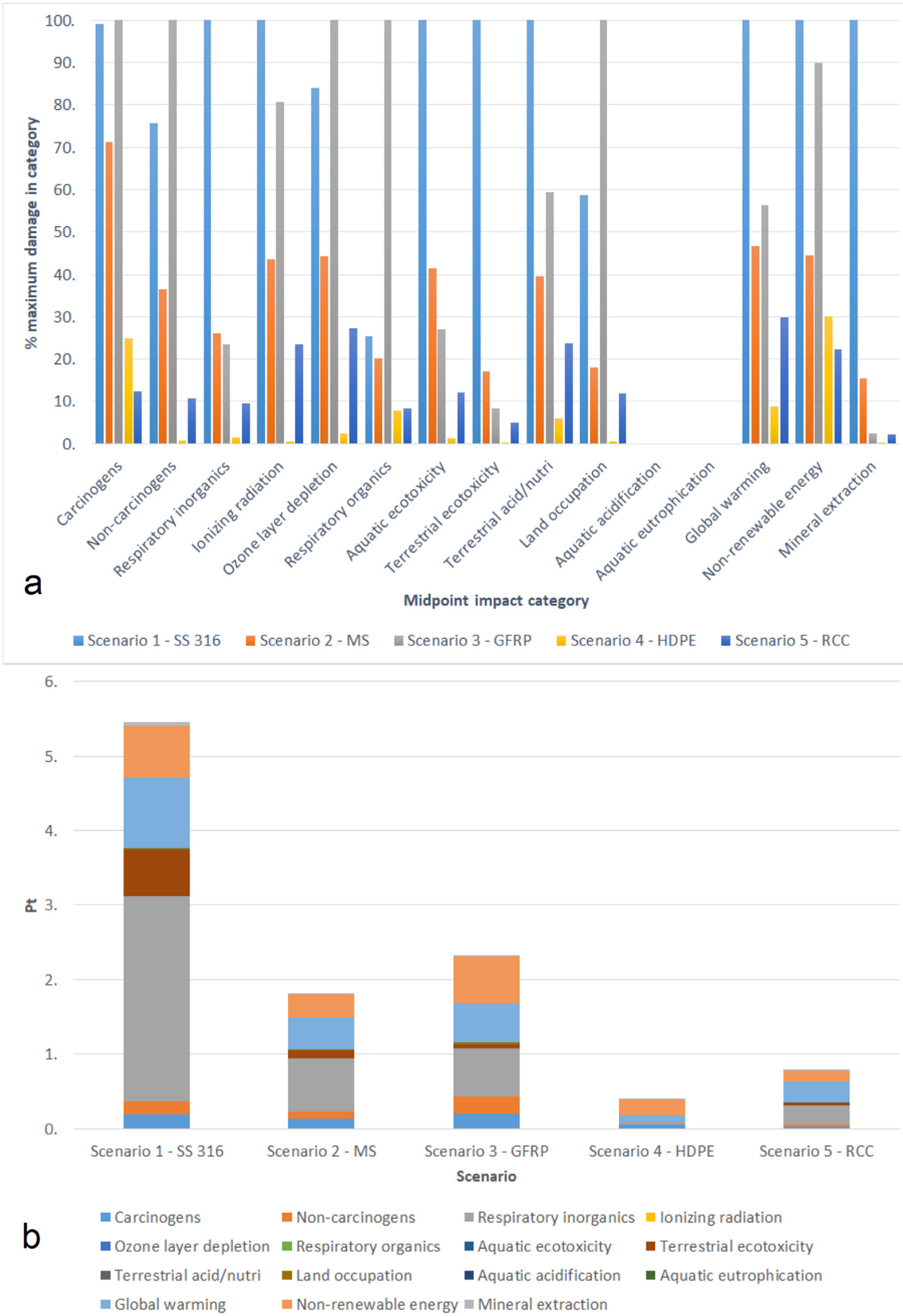

Fig. 2. Impact profiles across scenarios with regards to midpoint categories. (a.) Damage assessment by category. (b.) Single score. (Tiff file*2 columns)

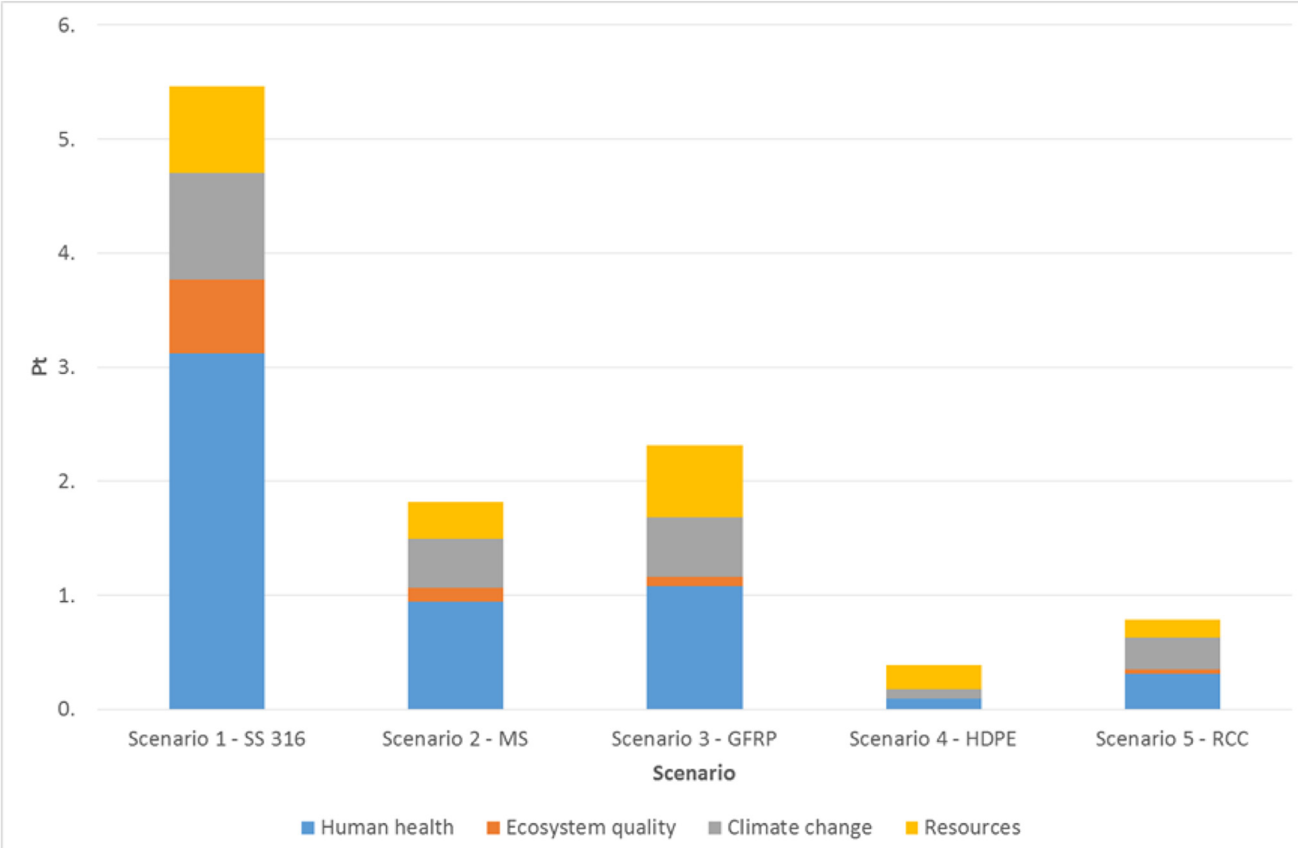

Fig. 3. Impact profiles of each scenario with regards to endpoint categories. (TIFF file*1.5 column).

(0.65 Pt). Scenarios 2, 3 and 5 followed the same trend but at lower levels of impact. In contrast, the highest impacted category for Scenario 4 was resources at 0.21 Pt due to high demand on non-renewable energy, followed by human health with 0.09 Pt and climate change at 0.08 Pt, all of which scored very low in comparison to alternatives.

These results are succinct with earlier work where Olmez et al. (2016) compared the effect of different processes in steel production on endpoint categories in an LCA. A common theme they identified across process and product scenarios in the cradle-to-gate analysis was that human health was the category incurring most impact, followed by climate change and resources. Similar results were reported by Shah et al. (2016) who conducted an LCA to compare the influence of three different materials (RCC, MS and PE) used to construct a 1,000 L water tank. As with the present study, Shah et al. (2016) found the PE product to contribute the least impact to human health, ecosystem quality and resource depletion categories, although they found RCC to incur higher impact costs than MS in all categories which is in contrast to the present study. Other contrasting results come from a study by Ibbotson and Kara (2013) who found resources to follow human health before ecosystem quality in a cradle-to-gate analysis of SS structural beams.

More recent work has shown that when the manufacturing phase is discounted from a cradle-to-gate analysis, it is the resources that would be most impacted due to the large demand on non-renewable energy to melt the iron (Liu et al., 2020). This implies it is the manufacturing phase that contributes most harm to human health which is understandable given the exposure of workers to carcinogens and fine particulate matter under 2.5_{um} ($PM_{2.5}$) during fabrication (Koponen et al., 1981; Sørensen et al., 2007) and supports earlier work that highlighted the greatest emission of airborne particulate matter during this phase (Stephens et al., 1998). However Ibbotson and Kara (2013) found the manufacturing phase to be responsible for only 7% of the contribution to human health, though this may be attributed to the limited fabrication required in beam production compared to heavily-welded items such as WWTP tanks.

3.2. Comparison of impact contribution from each scenario to human health

The human health damage category demonstrated both the largest portion of Scenario 1 with a score of 3.12 Pt and the greatest difference across scenarios. Scenarios 2 and 3 showed similar results which are approximately a third of Scenario 1 while Scenario 4 scored the lowest (0.09 Pt).

This investigation demonstrated that between 87.2 % and 98.5 % of all emissions impacting on human health are airborne as shown in Fig. 4. The remainder of emissions in Scenarios 1, 2 and 3 were found to be as a result of arsenic (As) emission to water with scores of 0.0003, 0.0002 and 0.0001 DALY respectively. Emissions into water of Scenarios 4 and 5 were negligible although previous work has shown that HDPE emits more waterborne metals than SS during the manufacturing phase (Stephens et al., 1998). Scenario 3 was the only scenario to demonstrate any substantial emission to soil which was again observed to be As emission.

Under Scenario 1, respiratory inorganics constituted 50.3 % of the total impact for this scenario and 88.1 % of the human health category. Regarding the emission of carcinogens under Scenario 1, scores were higher than Scenarios 2 and 3 (0.14-0.2 Pt) but to a greater extent with regards to Scenarios 4 (0.05 Pt) and 5 (0.02 Pt). These results suggested that the processing of HDPE was superior in terms of preserving human health compared to steel or GFRP products with SS demonstrating a significant risk to human health.

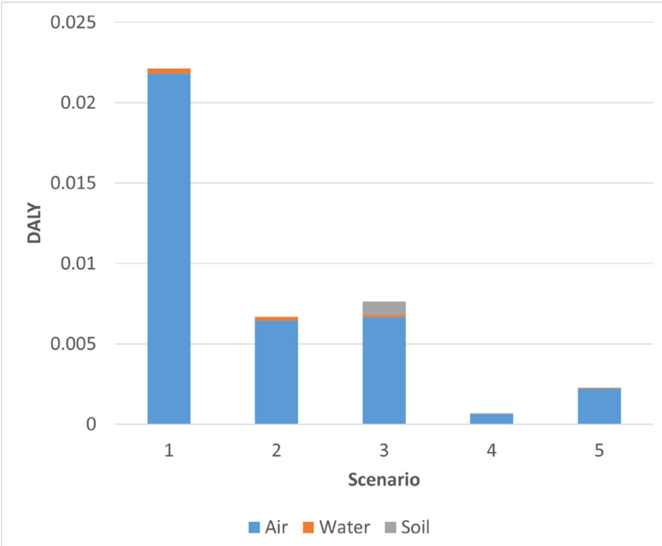

Fig. 4. Profile of emissions impacting to human health across scenarios. (TIFF files*1 column)

As shown in Fig. 5a, most of these scores are comprised of the air emission of fine particulate matter <2.5 μm ($PM_{2..5}$) with an elevated score of 2.2 Pt under Scenario 1 as expected (Amodio et al., 2013; Qing et al., 2015; Dunea et al., 2016; Panont et al., 2016). The $PM_{2.5}$ emissions of the next highest scenario, Scenario 2, were only 23.4 % of Scenario 1 while Scenario 4 was seen to produce negligible $PM_{2.5}$ emissions. Cr and Ni elements are known to be key markers of $PM_{2.5}$ emissions from SS production plants (Dunea et al., 2016). This is of notable concern because both Cr and Ni have been declared human carcinogens by several organisations including the World Health Organisation, International Agency for Cancer Research (International Agency for Research on Cancer 2012) and the US Department of Human Health Services (Department of Health and Human Services 2011). As displayed in Fig. 5b, Scenario 1 demonstrates considerable elevation of Ni in comparison to other scenarios, while Fig. 5c showed the same for Cr.

There are many pathways for Cr and Ni to impact human health within the life cycle of SS WWTPs. The International Agency for Research on Cancer (International Agency for Research on Cancer 1990) recognised SS workers to be amongst the three most at risk groups of people to exposure of Cr in its most toxic form. In SS fabrication plants, the emission of Cr and Ni mostly originate from the electrodes used and disperse via welding fumes (Koponen et al., 1981; Sørensen et al., 2007). As much as 3.5 % of welding fumes are known to consist of Cr, which has been shown to be the main source of toxicity in the fumes causing noticeable cell damage (White et al., 1979). Within alloy steel smelting units, particulate Cr is known to exist in the vicinity for long periods posing a substantial carcinogenic risk to workers despite the use of local ventilation systems (Mousavian et al. 2017).

Beyond the workplace SS production is widely known to be a considerable source of $PM_{2.5}$ and heavy metal emissions in the local environment (Rühling et al., 1992; Koleli and Halisdemir, 2005; Wang et al., 2010; (Amodio et al., 2013); Qing et al., 2015; Yang et al., 2015; Dunea et al., 2016; Panont et al., 2016). This poses significant health risks for the local population and environment, with the three main forms of exposure to heavy metal being ingestion, inhalation and dermal contact (Olawoyin et al., 2012). Kimbrough et al. (1999) highlighted in their review that inhalation of water-insoluble Cr particles released from industrial processes to be the foremost health risk of this metal to humans.

The health effects on populations due to proximity of SS production sites are apparent. A study by Dunea et al. (2016) investigated the distribution of $PM_{2.5}$ particles including Cr and Ni as key elements across a Romanian city in relation to the location of its metallurgic plants (including SS fabrication works), as well as the distribution of young children exhibiting respiratory problems. By doing so the authors identified a significant relationship between occurrence of respiratory problems and proximity to the metallurgic works, particularly in children aged between 2 and 7 years old. This is succinct with the findings of several other studies that found soil-deposited Cr in the environment to exceed acceptable thresholds and pose a notable carcinogenic risk to children (Wang et al., 2010; (Olawoyin et al., 2012); Qing et al., 2015; Wei et al., 2015).

As represented in Fig. 5d, Scenario 3 also demonstrated elevated emission of Cd compared to other scenarios that is another known carcinogen (International Agency for Research on Cancer, 1993). This is expected with glass production known to be a key source of airborne Cd emissions (Passant et al., 2002).

3.3. Comparison of impact contribution from each scenario to ecosystem quality

As with the impact to human health, ecosystem quality is also found to be most heavily impacted by air emissions as seen in Fig. 6, particularly Scenario 1 where air emissions total 96.2 % of the total emission types. In other scenarios this portion is seen to be less with air emissions accounting for between 52.9 % and 66.4 % for Scenarios 3 to 5, while Scenario 2 showed 90.3 %.

A study in China on the quality of water discharged from steel and iron industry identified a significant impact by way of water emissions from a steel production site (Sun et al., 2019). It is therefore surprising that ecological impact by way of water emission was found to be so low under Scenarios 1 and 2 as shown in Fig. 6. An explanation for this may be that the results of the present study do provide an accurate picture of the emission partition between air, soil and water and despite significance, water emissions may be negligible in contrast. Another explanation for this may be that the Impact 2002+ database fails to account for lesser point-source mitigation of wastewater from these industries in developing regions. Either way, as Sun et al. (2019) showed, the discharge of wastewater from steel industries in certain areas remain above acceptable standards and warrants attention.

Upon further investigation it was observed that these reduced portions were compensated by increased emissions to soil. This is expected as the main pathway for these toxicities to enter the ecosystem will be via airborne emissions subsequently being deposited on the soil (Koleli and Halisdemir, 2005; Panont et al., 2016). This is evidenced by the greater influence of soil emissions under Scenario 3 to diminishing ecosystem quality as shown in Fig. 6, and represents the elevated Cd emissions of this scenario. Cd is known to have considerable impact on microbial diversity in soil, soil fertility and nutrient cycling at higher levels of contamination that can lead to negative impact at higher trophic levels of the food web (Yao et al., 2003; Liao et al., 2005; Wu et al., 2018).

The present study reveals a strong asymmetry with a score of 8960 PDF*m^2*year under Scenario 1 as shown in Fig. 6. Scenario 2 demonstrated a reduction of 82 % when compared to Scenario 1, while Scenarios 3 and 5 incurred 86.8 % and 94.15 % less detriment to ecosystem quality respectively. The least impacting scenario was Scenario 4 which scored only 31.4 PDF*m^2*year which afforded a reduction in ecosystem degradation of 99.65 % compared to Scenario 1. These results showed that substantial benefits can be achieved in terms of preserving ecosystem quality if alternative materials to SS are utilized, with particular emphasis on the use of HDPE.

In order to understand better the cause of the high impact observed under Scenario 1, further investigation was warranted into the substances that formed the airborne emission profile. It was found that the airborne emissions were mostly comprised of only four heavy metals with Cr, Zn, aluminium (Al) and Cu accounting for 87.4 % of to-

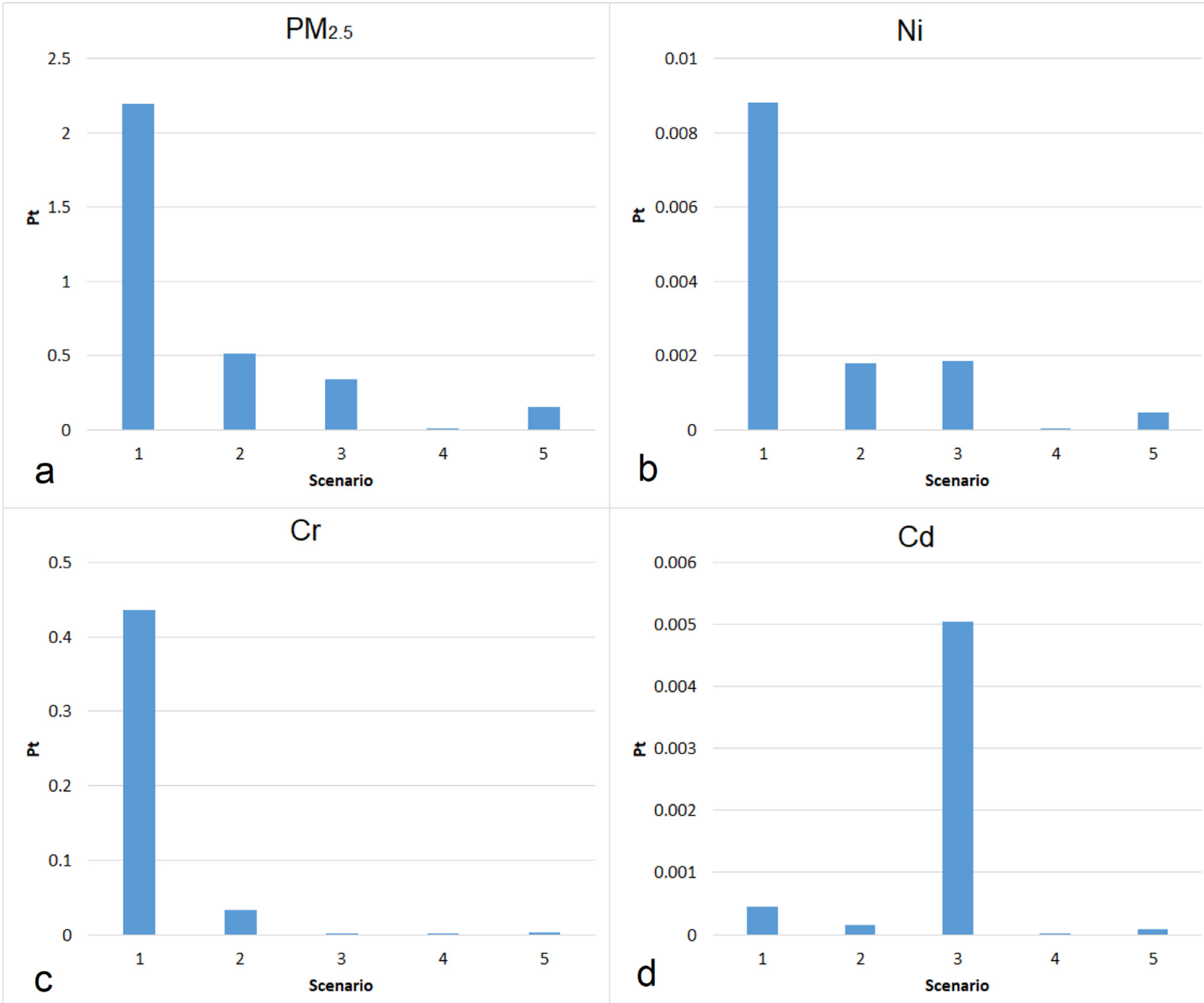

Fig. 5. Relative scenario contribution of $PM_{2.5}$ and key heavy metals to the human health impact category (a.) PM2.5, (b.) Ni, (c.) Cd, and (d.) Cr). (TIFF file*1.5 column).

tal emissions and 93.1 % of airborne emissions. Of these, Cr was observed as the prominent element. It is evident that much of the impact to ecosystem quality is attributed to the emission of these elements. Of these heavy metals, Cr, Zn (as well as Ni) are known to be highly detrimental to organism function, i.e. plants (Kloke et al., 1984), and their bioaccumulation is now present throughout many aspects of the ecosystem (Palaniappan and Karthikeyan, 2009; Orlowski et al., 2014; Orisakwe et al., 2015; Chen et al., 2018; Kazi et al., 2019; Usman et al., 2019).

3.4. Comparison of impact contribution from each scenario to climate change

Considering three of the scenarios, climate change was the endpoint category that demonstrated the second highest contribution to total impact after human health. Within the midpoint categories it was found that only global warming contributed to the endpoint category (climate change). Inspection at the substance level identified CO_2 production from fossil fuel use to be the primary contributing factor. As shown in Fig. 7, Scenario 1 produces a total of 8.77 tonnes of CO_2 through fossil fuel use compared to Scenarios 2 and 3 that produced about half (4.07 and 4.46 tonnes respectively) the value of Scenario 1. Scenario 5 produced a total of 2.66 tonnes CO_2 while Scenario 4 again produced the lowest of 0.73 tonnes CO_2 in total.

In terms of climate change, the use of HDPE in place of SS could reduce CO_2 emissions from fossil fuels by 91.7 %. Savings of approximately 50 % could be achieved by employing either MS or GFRP in place of SS, while savings of 69.7 % would be made through the use of RCC. These findings are supported by Machado et al. (2007) who identified a potential reduction of 1 % in both CO_2 emissions when the steel in an activated sludge (AS) reactor was replaced with HDPE during investigation into improved sustainability of small WWTPs. They further identified potential reductions of CO_2 emissions of 1 % when concrete was replaced by HDPE in an Imhoff tank.

In terms of water and sewage pipework other work has compared material impact on climate change, although SS is generally not used in these roles. For example, Recio et al. (2005) compared the contribution of CO_2 emissions during the production of water pipes using several materials. The authors identified HDPE to be the lowest contributor compared to different plastic types, concrete and the highest contributor,

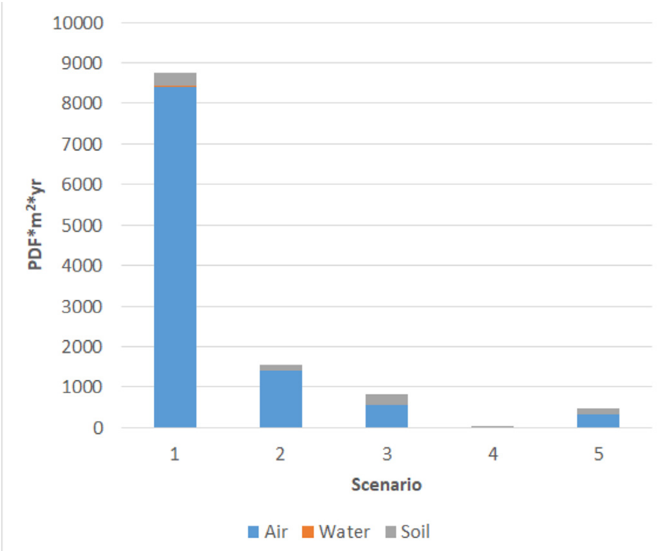

Fig. 6. Profile of emissions impacting ecosystem quality across scenarios. (TIFF file*1 column).

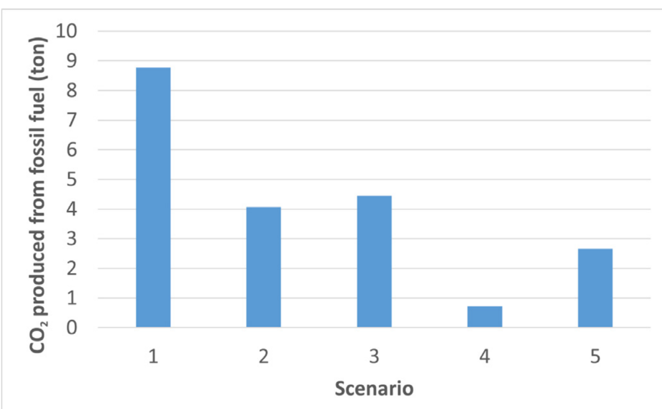

Fig. 7. Relative CO_2 profiles of each scenario due to fossil fuel consumption. (TIFF file*1 column).

ductile iron (Recio et al., 2005). Another study found HDPE to contribute less greenhouse gases (GHG) than concrete piping with cast iron the highest (Kim et al., 2012), while other plastic types such as PVC contributed the least GHG overall.

Previous studies indicate that the use of plastic polymer may contribute as much as 10–26 times as much GHG than concrete (Venkatesh et al., 2009; Viñolas 2011). This was supported by Du et al. (2013) who found RCC to hold a lower global warming potential (GWP) than HDPE in contrast to the present study, while cast iron and ductile iron were substantially higher. Other research has suggested alternative production pathways in the PE industry are gaining prominence and are able to further reduce GHG emission of HDPE by 7–33 % (Yao et al., 2016).

In comparing the output of GWP and embodied energy by production of structural steel and RCC, the latter was found to incur less contribution in a recent study (Kua and Maghimai, 2017), however this contribution could be heavily reduced by including a higher share of secondary steel in the production process. The authors found the results could be reversed when a new emerging technique were employed during production known as “near net shape casting” whereby the metal is cast to a shape similar to the finished product thus avoiding the need for reheating it before rolling. They proposed a saving of almost 5 MJ/kg of steel was achievable. With EE and GWP being so closely related during production (Recio et al., 2005; Kua and Maghimai, 2017), widespread adoption of this technique could indicate a future paradigm shift relating to the relative contribution of steel types to climate change when compared to alternative materials.

Other work has demonstrated that energy consumption and CO_2 emission could be reduced by up to 70% if virgin production was produced through sole use of scrap material (Johnson et al., 2008), however the authors also highlight that limited availability of scrap metal would make this scenario unfeasible. Reck and Graedel (2012) proposed the best ways to improve recycling rates of metal which includes better systems for collection of scrap, improved recycling design and more widespread employment of modern recycling techniques.

3.5. Comparison of impact contribution from each scenario to resource depletion

Within the resource depletion category, several resources demonstrated considerable asymmetry. Fig. 8a shows Scenario 1 to place particular pressure on Cr reserves compared to all other scenarios, while Fig. 8c demonstrates a near-identical demand on Ni reserves across scenarios. This is expected as these are key minerals used in the production of austentic SS. In contrast, Molybdenum (Mo) is only present in certain grades of austentic SS alloys and at less than 3% (Martins et al., 2014). However, Mo is commonly used at higher quantities as a strengthening agent in low-alloy steels such as MS and reinforcing steel (Yellishetty et al., 2011; Uranga et al., 2020) which explains the disproportion observed in Fig. 8b compared to other scenarios.

Zn demand was substantially higher in Scenario 3 as shown in Fig. 8d, requiring 0.793 kg compared to the lowest demand of 0.01 kg under Scenario 4 and the second highest at 0.326 kg under Scenario 1. This is of higher concern as Zn has been identified as a mineral requiring an immediate reduction in extraction rate of 82 % if sustainability is to be achieved (Henckens et al., 2014). Henckens et al. (2014) also proposed that a 63 % reduction in Cu was required, which demonstrated similar demand to Zn with Scenario 1 requiring 0.625 kg compared to 0.171 kg, 0.375 kg, 0.001 kg and 101 kg for Scenarios 2 – 5 respectively. It is clear that the use of alternative materials in construction can afford substantial savings on declining mineral reserves when compared to current practices. This was supported by Machado et al. (2007) who identified a potential reduction of 1% in abiotic depletion when steel was replaced with HDPE in an AS reactor and as much as 5% reduction when concrete was replaced with HDPE in an Imhoff tank.

A study conducted by Yellishetty et al. (2011) to investigate the role of the steel industry on abiotic resource depletion. Their synthesis was that the overall impact on abiotic resource depletion derived through LCA has not been holistic in its assessment, failing to account for the socio-economic impact that it incurs particularly in developing nations. These nations are particularly vulnerable to future economic deterioration due to their heavy export of mineral stocks for short term gain. While underdeveloped nations currently export them at an unsustainable rate to improve their current economic situation, they diminish future availability for their own national development as demonstrated by Africa, South America and large parts of Asia (Yellishetty et al., 2011). Recapture of these minerals through recycling commands potential, however significant hurdles first need to be addressed including overcoming the limited traffic of recyclable waste needed to achieve economic feasibility of these processes (Yellishetty et al., 2011).

3.6. Consideration for the end-of-life phase in the present LCA

While this study has excluded consideration for end-of-life (EOL) options for each material scenario, this will influence the relative impact ranking of each material (Rives et al. 2010; Hottle et al., 2017). In comparing the life cycles of SS and HDPE vehicle fuel tanks, Stephens et al. (1998) found HDPE to outperform SS in most categories

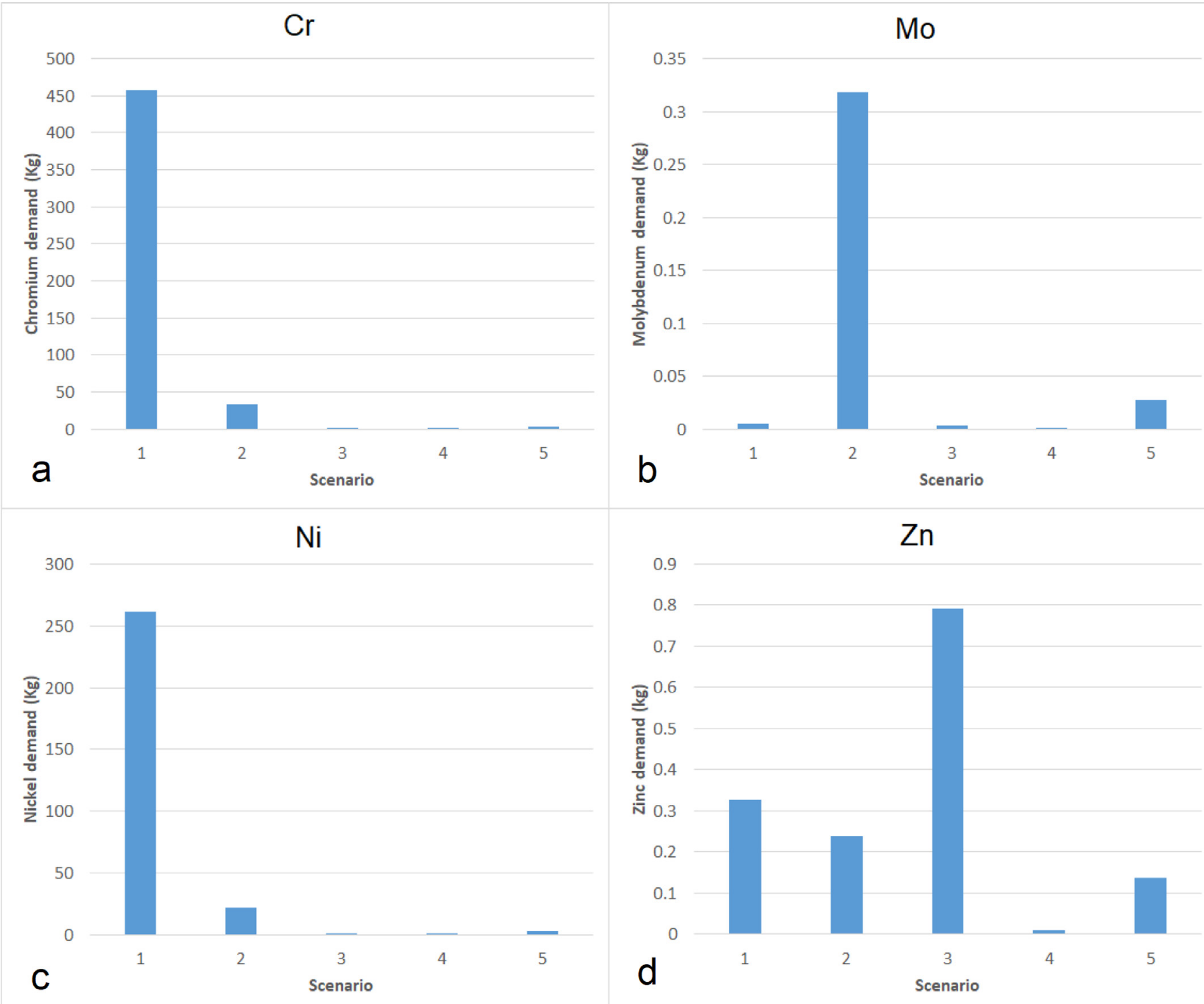

Fig. 8. Relative demand on key finite resources (a.) Cr, (b.) Mo, (c.) Ni, (d.) Zn). (TIFF file*1.5 column).

but did find SS to be the most environmentally-friendly over the EOL phase due to the ease of recycling which was unachievable with HDPE at that time. The recycling of plastic is becoming more commonplace, but remains low in developing countries (Sadat-Shojai and Bakhshandeh, 2011) while the environmental benefits of recycling are limited by the need to ship waste overseas (Hottle et al., 2017).

GFRP has been deemed generally unrecyclable due to a number of limitations that include the high energy demand required in the recycling process (Correia et al., 2011; Shuaib and Mativenga, 2016). Shuaib and Mativenga (2016) did infer that the energy demand could be heavily reduced if enough waste was available. Under this scenario Shuaib and Mativenga (2016) proposed that the production of mechanically recycled GFRP would demand only 0.17–1.93 MJ/kg compared to the production of virgin glass fibres requiring 13–54 MJ/kg. They also showed mechanical recycling to be the superior practice in terms of energy demand when compared to alternative methods (Shuaib and Mativenga, 2016).

Until sufficient material traffic is available to make recycling economically and energy feasible, other options for recycling should be explored that could lead to improved EOL profiles for these materials. While GFRP can be reduced to fine debris and recycled in non-structural concrete, it has been shown to diminish performance which would limit its application (Correia et al., 2011). Previous studies suggest that it offers advantage in marine engineering (Zhang et al., 2019), although the application is shape dependent and more suited to the repurposing of pipework rather than tanks. Due to the high anti-corrosion properties of GFRP, these tanks could help promote biodiversity as artificial habitat in the marine environment providing a more feasible form of recycling (Santos et al., 2011; Lokesha et al., 2013; Sreekanth et al., 2019).

Further complexities arise when processes are known to influence multiple categories. For instance, a recent study by Nguyen et al. (2020) concluded that higher GHG release from HDPE would be expected when incinerated or mechanically recycled compared to landfilling. However, a study by Sangwan and Bhakar (2017) showed that landfilling HDPE may contribute heavily to the human health category due to the emission of vanadium (V) ions. Consideration for such trade-offs will need to be incorporated into the design phase of small WWTP if environmental burden is to be effectively minimized.

4. Conclusion

This study is considered a first step towards identifying alternative materials that may help reduce environmental impact of a packaged wastewater treatment systems. The study showed that the use of SS as the primary material incurs substantial environmental burden during the early life stages across 9 of the 13 midpoint impact categories investigated and all endpoint damage categories. The most impacted midpoint categories when SS was used were respiratory inorganics, mineral extraction and both aquatic and terrestrial ecotoxicity categories. From overall perspective, HDPE was identified as the least impacting material offering a potential early-life impact reduction of 93 % compared to SS. Other materials such as RCC, MS and GFRP were also observed to offer considerably less environmental impact than SS.

Further study is recommended to investigate the longevity of different materials in the role of a wastewater treatment asset on a longer term LCA to generate a more comprehensive comparison, particularly where system assemblies and subassemblies need to be replaced over the study period. Focus should also be given to other life stages such as transportation, installation and disposal to see how these are affected under alternative material scenarios.

A consideration to the findings of this paper may help to contribute towards the global aspiration of achieving the sustainable development goals by 2030.

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Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.envadv.2021.100065.

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